Hatches Harbor Salt Marsh Restoration: 2002 Annual Report

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INTRODUCTION

In cooperation with the Town of Provincetown, Federal Aviation Administration and Massachusetts Aeronautics Commission, the National Park Service (NPS) has been incrementally restoring tidal exchange to the diked portions of Hatches Harbor since March of 1999. The overall objective of this project is to restore native salt marsh functions and values to the tide-restricted wetland to the extent possible without compromising safety at the Provincetown Municipal Airport.

After an hydrodynamic assessment, large culverts were installed through the Hatches Harbor Dike by the NPS in the winter of 1998-99 to accommodate increased tidal flow. These gated culverts have been opened in small increments each year (Table 1) to ensure Airport safety from flooding and to control and adaptively manage ecosystem response. Cape Cod National Seashore (CCNS) staff and cooperators have monitored system response intensively since 1999, with base line data collected in 1997 before new culvert construction.

This reports on physical and ecological monitoring undertaken in 2002 and summarizes progress towards habitat restoration. Monitoring has included tide heights, sedimentation, sediment-water quality, wetland vegetation, nekton (fin-fish and decapod crustaceans) and mosquitoes within both natural (unrestricted) and diked portions of the Hatches Harbor flood plain (Fig. 1). The 2002 sampling included all but nekton; see the 2000-2001 Annual Report for 1997-1999 nekton data. We plan to sample nekton again in 2003.

Table 1. Recent history of incremental culvert gate openings at Hatches Harbor.

Years	Number of open culverts	Dimensions of opening	Opening area (m^2)
Pre-1999	1	2-ft ID old round culvert	0.29
1999	2	2.13 m wide X 0.10 m high	0.42
2000	4	2.13 m wide X 0.10 m high	0.85
2001-2	4	2.13 m wide X 0.40 m high	3.41

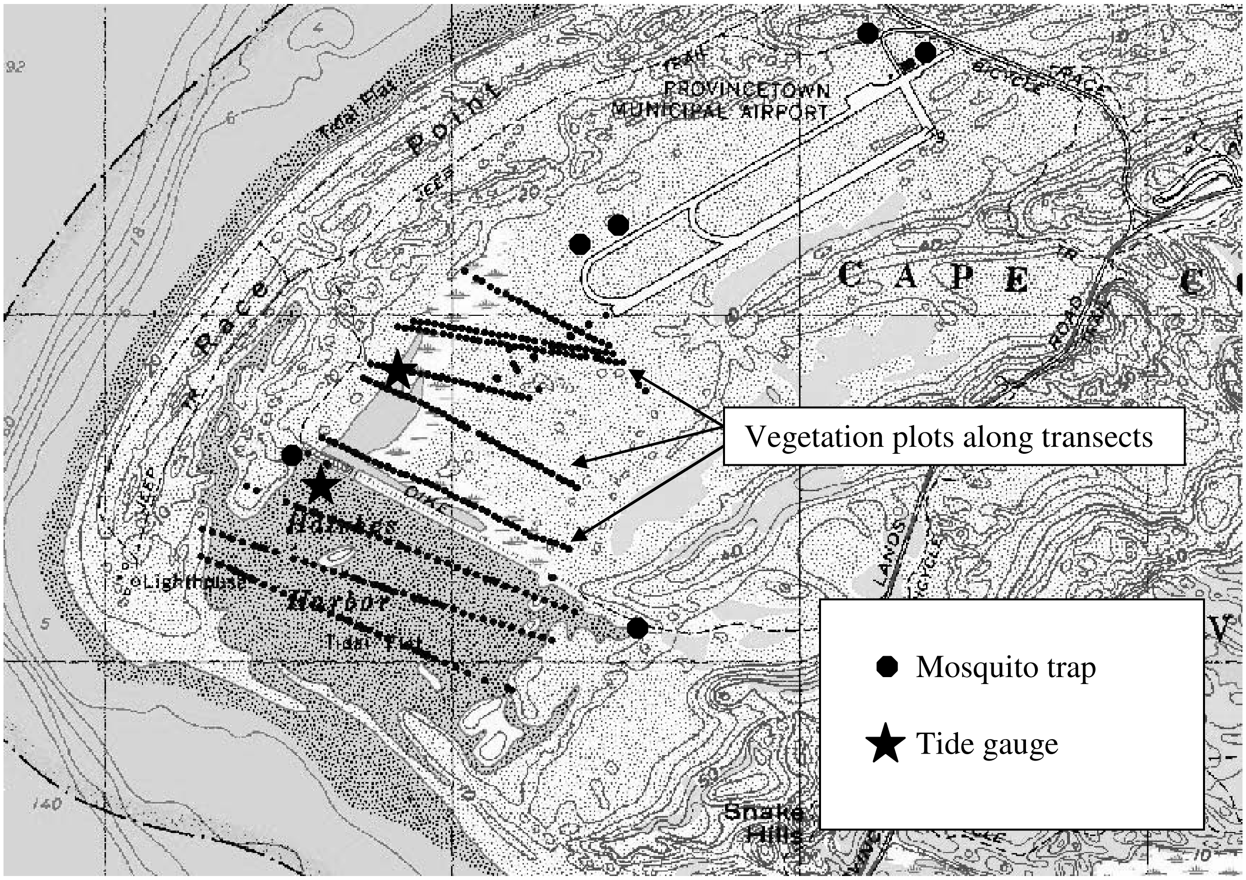


Figure 1. Hatches Harbor salt marsh showing locations of tide gauges, transects for vegetation and porewater sampling, and mosquito trapping stations.

TIDE HEIGHTS

Data loggers (YSI UPG6000) are installed on stable mounts on both sides (seaward and landward) of the dike to record water depth and salinity in the principal tidal creek over typically one-month deployments. The seaward station is about 10 m from the dike; the landward station is about 300 m above the dike in the main creek draining the restricted marsh. Depth transducers are referenced to m-MSL by differential leveling from local benchmarks whose elevations are given in ft-MLW; ft-MLW units are converted to m-MSL by subtracting half the tidal range for Provincetown (9.1 ft/2 = 4.55 ft) and dividing by 3.28 (ft/m). Data are recorded at 15-min intervals.

The Hatches Harbor culverts were intentionally designed to be low and wide after modeling showed that restricting culvert height effectively dampened storm tides, while a wide profile provided efficient ebb-tide drainage (Roman et al. 1995). Recent data demonstrate how well the culvert system actually works in the field: seaward tide heights of 2.3 m-MSL on 6 October 2002 were reduced to 1.7 m-MSL by passage through the dike; meanwhile standard high tides were much less affected (Fig. 2.) Thus, as the model predicted, standard high tides and tidal range are being maximized to favor the restoring salt marsh while extreme high tides are being buffered out to protect Airport facilities.

Figure 2. Hatches Harbor tide height data from 11 September to 16 October 2002 showing the dampening effect of the dike and culvert system on extreme high tides.

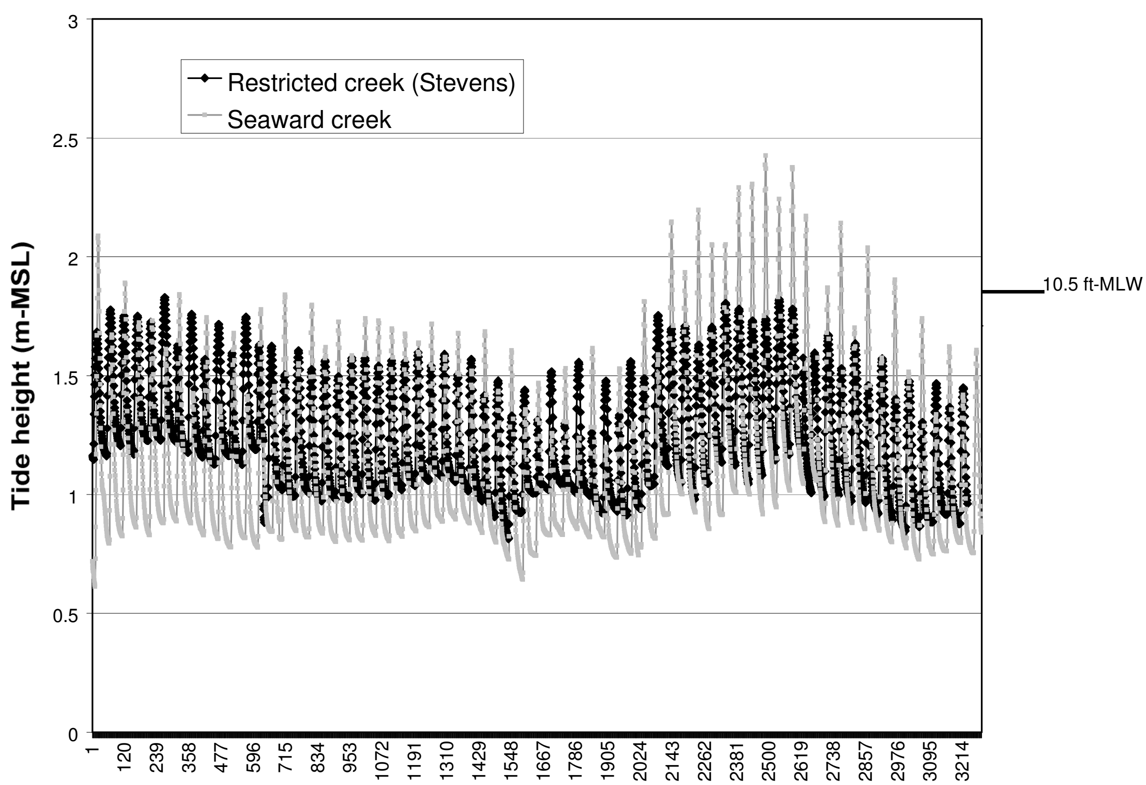


Table 2 summarizes all verified tide height data. Figure 3 presents mean high and low tides from 1998 (pre-restoration) to the present relative to a typical restricted marsh profile, i.e. Transect 2 running perpendicular to the main creek and about 300 m from the dike. In general, high tide elevation has increased as expected with increased culvert opening. Average high tides did not flood the marsh surface before the new culverts were installed (1998). The initial 1999 opening of two of the new culverts 0.10 m raised mean high tide about 6 cm. Notably, this replacement of the old circular 2-ft (0.61-m) opening with the lower-profile and wider new culverts caused mean low to decrease 4 cm; thus, marsh drainage during low tide was enhanced. Opening all four culverts 0.10 m high in 2000 resulted in little change in either mean high or mean low tides.

The 2001 opening of all four culverts to 0.40 m has resulted in a substantial increase in tide heights, with only a modest increase in mean low tide. This translates to a large increase in the areal extent of wetland flooding during high tides, while still maintaining good marsh drainage during low tides. The 18 cm increase in mean high between 2001 and 2002 (Fig. 3), when the culvert gates were unchanged, is unexplained: it may have been due to 1) differential tidal forcing from Cape Cod Bay (undocumented) and/or 2) a change in creek morphometry allowing more water to enter the upper creek system. Tidal range in the restoring marsh is now about 60% that of unrestricted Hatches Harbor (Table 2).

Figure 3. Typical wetland profile and recent history of mean high and low tide heights in the restoring Hatches Harbor wetland.

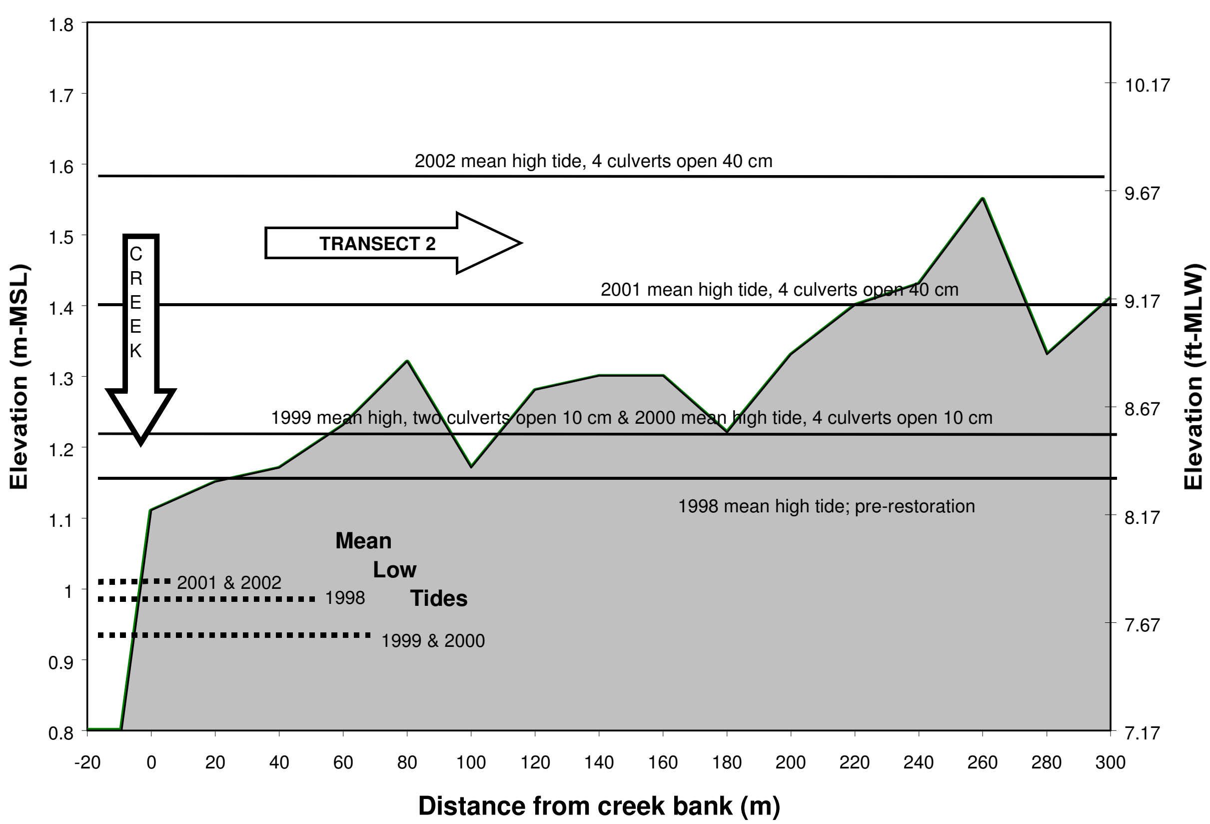
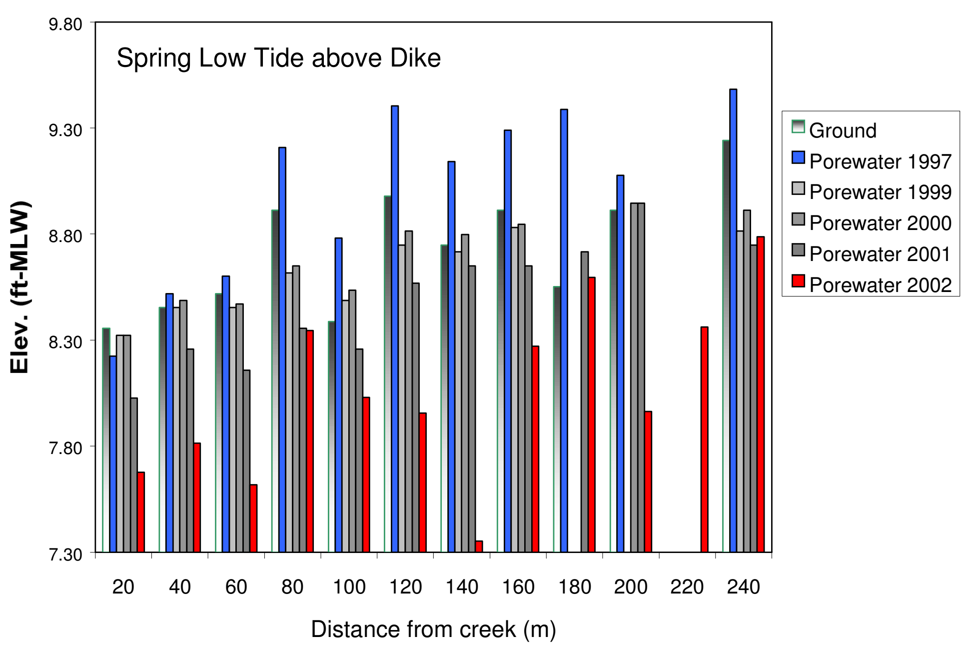


Table 2. Summary of tide height data from restricted (landward) and unrestricted (seaward) portions of the Hatches Harbor main creek. Elevations are m-MSL.

Landward of dike	Sampling interval	Mean high	Mean low	Tide range (m)
2-ft culvert w/o flap	3/30-5/6/98	1.16	0.99	0.17
2 culverts open 10 cm	May-Jun 1999	1.22	0.95	0.27
4 culverts open 10 cm	2/14-2/28/01	1.21	0.93	0.28
4 culverts open 40 cm	5/31-6/21/01	1.38	0.96	0.41
	4/14-5/3/01	1.40	1.03	0.37
	9/11-10/16/02	1.58	1.02	0.56
Seaward of dike	5/19-6/4/99	1.55	0.92	0.63
	9/11-10/16/02	1.77	0.84	0.93

To represent hydroperiod within the vegetated wetland, tide heights are also monitored over a spring-neap tide cycle during Aug-Sep within the wetland peat along Transect 2. A critical variable affecting wetland plant vigor is the depth of dewatering and consequent root-zone aeration during the growing season. Prior to tidal restoration, the restricted wetland surface often remained flooded even through the low tide period; with enlarged culverts, low-tide porewater levels have decreased substantially (Fig. 4). This, together with increased salinity and low sulfide (see below) should encourage the re-establishment and increase the vigor of salt marsh plants.

Figure 4. Low spring tide porewater levels along Transect 2 in the restoring Hatches Harbor salt marsh, along with land surface elevation, 1997-2002



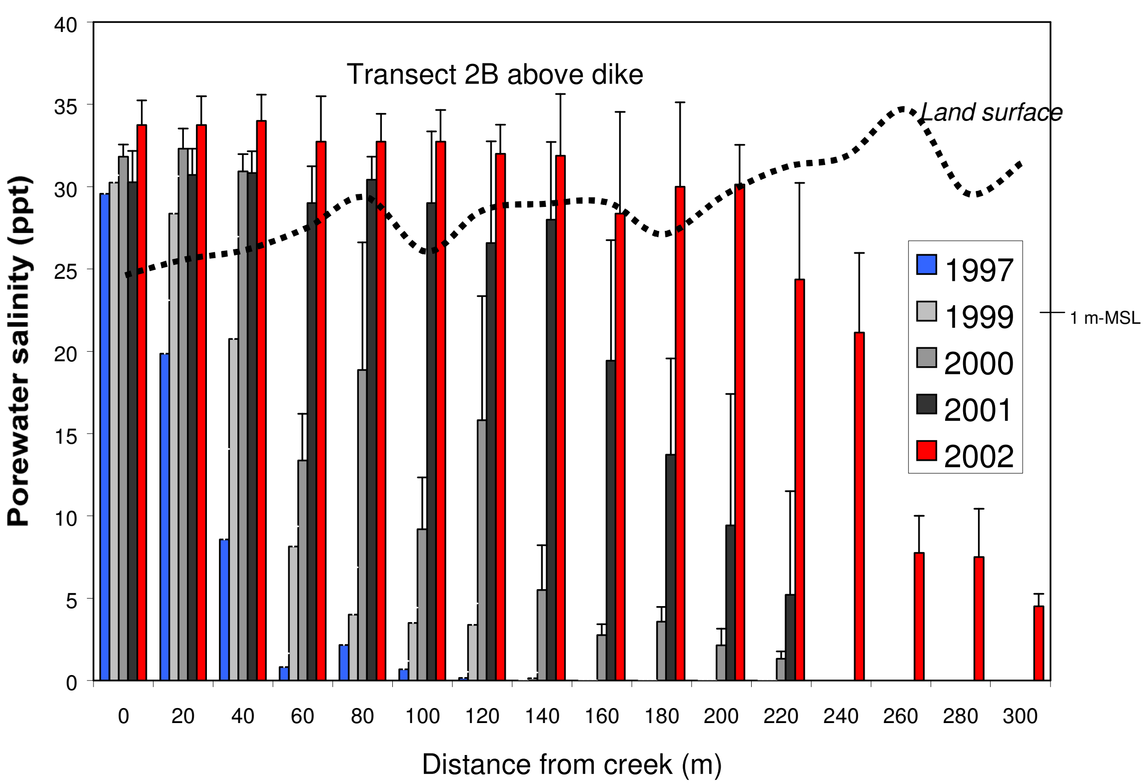
SEDIMENT-WATER QUALITY

Along with water levels, salinity and sulfide have been monitored before restoration (1997) and annually since 1999 in 10-cm-deep porewater in the vegetation plots of Transect 2; see the 2000-2001 Annual Report for methods.

The objective of this monitoring is to document root-zone conditions for emergent wetland plants. It is anticipated that increased salinity will stress *Phragmites* and freshwater wetland vegetation and thereby favor the re-establishment and competitive advantage of salt marsh grasses. Increased seawater supply should also increase sulfide generation, but only if sediments remain waterlogged throughout the tide cycle.

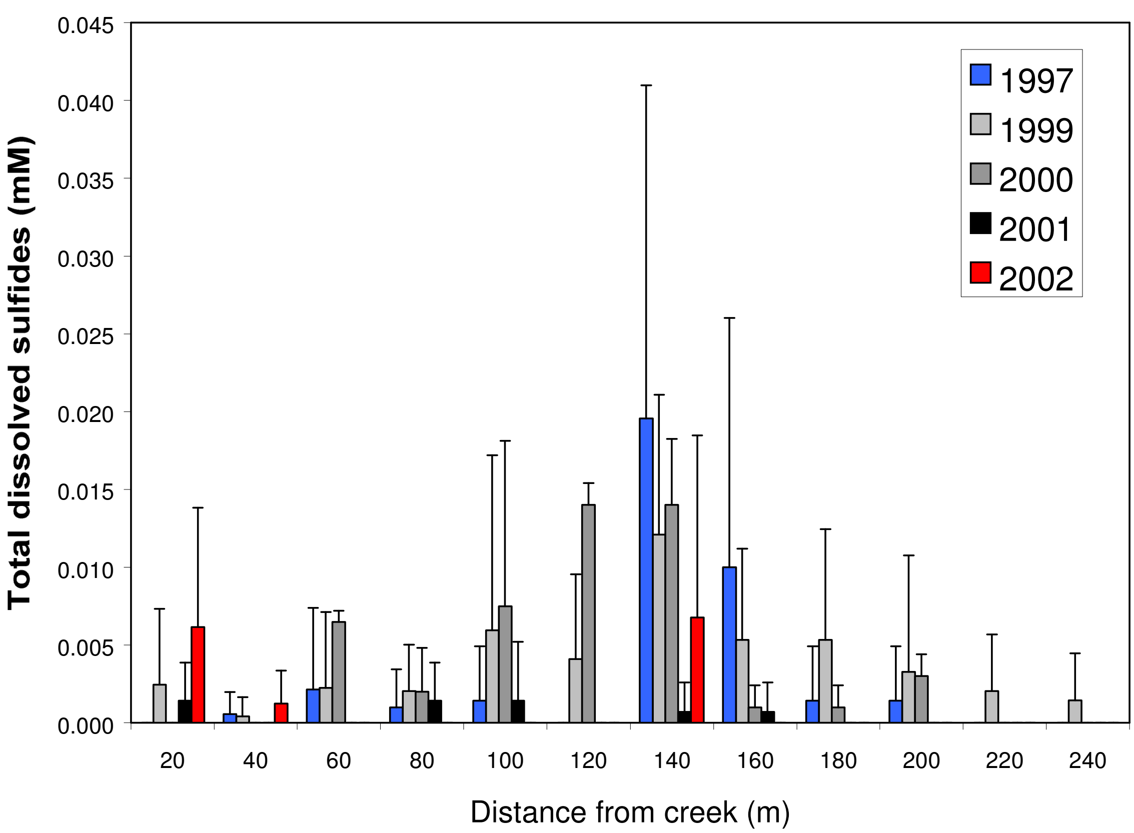
Figure 5 presents mean salinity of typically 7-9 low-tide observations through the spring-neap period in Aug-Sep of each year. Water of salinity > 20 ppt now penetrates the wetland 240 m from the creek bank, with 30-33 ppt salinity up to 200 m.

Figure 5. Mean spring-neap salinity in 10-cm-deep porewater along Transect 2 in the restoring Hatches Harbor salt marsh, 1997-2002.



Total sulfides in the root zone have been monitored along with salinity (Fig. 6). Sulfides have remained consistently below 20µM throughout this wetland, or about 100 times lower than many typical Cape Cod Bay salt marshes. This is expected with low low-tide levels and an unusually highly permeable, low-organic-content peat that promotes the drainage of water and entry of air into the root zone at low tide. Tidal restoration has not increased waterlogging and, thus, sulfide concentrations, which remain about two orders of magnitude below that likely to stress salt marsh plants. These observations, plus the fact that sulfide analysis is time-consuming and generates hazardous waste, compel us to suggest that routine sulfide monitoring at vegetation plots be terminated, except to address specific research questions.

Figure 6. Mean spring-neap sulfide concentration in 10-cm-deep porewater along Transect 2 in the restoring Hatches Harbor salt marsh, 1997-2002.



VEGETATION PLOTS

With restoration of tidal flow to the Hatches Harbor salt marsh, it is expected that vegetation of the tide-restored marsh will change dramatically from a *Phragmites australis* and shrub-dominated brackish wetland to a *Spartina*-dominated marsh typical of southern New England. Throughout the northeastern US, tides are being restored to tide-restricted marshes and conversion of vegetation to *Spartina* marsh is being successfully realized (e.g., Burdick et al. 1997, Roman et al. 2002, Warren et al. 2002).

At Hatches Harbor a long-term vegetation monitoring program has been established to track the vegetation response to tidal restoration. The purpose of this report is to summarize changes in the vegetation community that have occurred after 3 and 5 years of tide restoration.

Methods

The fundamental study design follows a BACI (before, after, control, impact) approach, with sampling of vegetation <u>before</u> tidal restoration and <u>after</u> restoration. The portion of the marsh located downstream of the Hatches Harbor marsh, referred to as the tide-unrestricted marsh, serves as a <u>control</u>. In the case of our BACI design, restoration of tidal flow is the <u>impact</u>. With this approach it is possible to make several comparisons aimed at determining, with statistical certainty, the response of vegetation to tidal restoration. The following comparisons are presented in this summary report;

- 1997 Unrestricted Marsh vs. 1997 Tide-restricted Marsh
  This represents conditions before any tidal restoration and will document the effect that tidal restriction has had on marsh vegetation.

- 2000 Unrestricted Marsh vs. 2000 Tide-restored Marsh
  2002 Unrestricted Marsh vs. 2002 Tide-restored Marsh
  This sampling scheme represents 3 and 5 growing seasons of restored tidal flow (2000 and 2002). With these comparisons, continuing for the long-term (e.g., 10 yrs of tide-restoration, 15 yrs of tide-restoration) it becomes possible to evaluate the trajectory or direction of vegetation change. Over time, it is assumed that the unrestricted and restoring marshes will become more similar.

- 1997 Tide-restricted Marsh vs. 2000 Tide-restored Marsh
  1997 Tide-restricted Marsh vs. 2002 Tide-restored Marsh
  This directly evaluates the response of the restoring marsh to re-introduced tidal flow. Overtime, if restoration is successful there will be increases in typical salt marsh species and decreases in brackish and woody species within the tide-restored marsh.

- 1997 Unrestricted Marsh vs. 2000 Unrestricted Marsh
  1997 Unrestricted Marsh vs. 2002 Unrestricted Marsh
  2000 Unrestricted Marsh vs. 2002 Unrestricted Marsh

It is important to document vegetation changes of the control marsh overtime. If after tidal restoration, vegetation changed within the tide-restored marsh, but the control marsh vegetation did not change, then it can be suggested with some certainty that the changes in the tide-restored marsh were due to increased tidal flow and not some other factors. Over the long-term (decades), it is expected that vegetation of the control marsh will change in response to sea-level rise, altered hydrology, and other factors; however, vegetation changes in the tide-restored marsh are expected to occur more rapidly.

A detailed presentation of field and data analysis methods are provided in the Seashores salt marsh vegetation monitoring protocol (Roman et al. 2001) and will only be briefly described here. Permanent 1 m^2 vegetation plots were established in a stratified-random manner throughout the unrestricted marsh and tide-restricted marsh and initially sampled in summer 1997. Follow-up monitoring occurred in summer 2000 and summer 2002. In each plot, vegetation species composition and abundance (cover) were estimated by the point-intercept method. The vegetation plot data were analyzed by Analysis of Similarity (ANOSIM), a non-parametric procedure aimed at comparing the similarity (or dissimilarity) between groups of community data (e.g., tide-restricted vs. tide-restored). A multivariate community ordination technique, detrended correspondence analysis, was also used to document vegetation community changes. The analyses were performed on 37 permanent plots in the unrestricted marsh and 74 tide-restricted/tide-restored plots. It is noted that when the vegetation plots were originally established in 1997, several plots in the tide-restricted marsh included higher elevation remnant dune areas, dominated by *Deschampsia flexuosa*. These plots are not likely to be influenced by increased tidal flooding because of their elevation, and thus, have been eliminated from the analyses.

RESULTS AND DISCUSSION

General Species Composition: Just five marsh plants are encountered within the *Spartina alterniflora* dominated unrestricted salt marsh (*S. alterniflora, S. patens, Salicornia sp., Limonium nashii, Sueada maritima*). Two species of brown macroalgae also contributed substantially to cover within the unrestricted marsh (*Ascophyllum nodosum* and *Fucus vesiculosus*). Over 80 species of vascular plants were found within the tide-restricted/tide-restored marsh.

Community Analysis: Following the BACI study design, numerous comparisons were made (Table 3). As expected, vegetation of the 1997 unrestricted and 1997 tide-restricted marsh was significantly different. After 2 years of tidal restoration, there were no significant changes detected, probably because of the small initial openings of the new culverts. By 2002, however, after 5 years of tidal restoration, significant changes were detected in the tide-restored marsh. [Note that despite these changes, even after 2 and 5 years of tidal restoration, vegetation above and below the dike structure remained distinctly different.] Vegetation in the unrestricted control marsh remained similar among sampling years, suggesting that the plant community changes in the tide-restored marsh were related to tidal restoration activities and not some other factors (Table 3).

Table 3. Results of one-way ANOSIM tests for pair-wise comparisons of vegetation data. Bonferroni adjusted alpha = 0.05/7 comparisons = 0.007.

Comparisons	***p* Value**
Unrestricted vs. tide-restricted or restored	
Unrestricted 97 vs. Restricted 97	0.001
Unrestricted 00 vs. Restored 00	0.001
Unrestricted 02 vs. Restored 02	0.001
Tide-restricted vs. Tide-restored	
Restricted 97 vs. Restored 00	0.04 (NS)
Restricted 97 vs. Restored 02	0.001
Unrestricted Control Marsh	
1997 vs. 2000	0.36 (NS)
1997 vs. 2002	0.17 (NS)

Aside from knowing that the vegetation community of the tide-restored marsh changed after 5 years of restored tidal flow, it is of interest to know which plant species contributed most significantly to this difference. Table 4 shows that the dissimilarity in vegetation from 1997 tide-restricted to 2002 tide-restored conditions was mostly attributed to a decrease in brackish marsh species, such as purple loosestrife (*Lythrum salicaria*) and soft rush (*Juncus effusus*), as well as some woody vegetation (e.g., blackberry, *Rubus* and bayberry, *Myrica pensylvanica*). It is not surprising that with increases in soil salinity and flooding duration, these species are beginning to die-back. It is also noted that the salt marsh species, black grass (*Juncus gerardii*), also deceased in cover with tidal restoration. Black grass typically grows at the upland border of marshes or at higher elevations and is intolerant of prolonged high soil salinities.

Table 4. Individual cover types contributing most to dissimilarities noted between the 1997 tide-restricted and 2002 tide-restored marsh. Cover class data are presented as the mean of five ordinal ranked classes (1 = <1-5%, 2 = 6-25%, 3 = 26-50%, 4 = 51-75%, 4 = >75%). Cover types cumulatively contributing up to 70% of the dissimilarity are presented.

Cover Type	Mean % Cover Class		% Dissimilarity
	97 Restricted	02 Restored	
Spirea alba	0.94	0.16	9.9
Lythrum salicaria	0.90	0.14	9.4
Rubus sp.	1.70	0.96	8.9
Myrica pensylvanica	1.22	0.48	8.9
Euthamia tenuifolia	0.86	0.19	7.3
Juncus effusus	0.82	0.23	5.6
Carex sp.	0.73	0.16	5.3
Myrica (DEAD)	0.13	0.67	5.1
Holcus lanatus	0.79	0.27	4.4
Juncus gerardii	0.84	0.34	4.1

One of our restoration objectives is for the vegetation of the tide-restored marsh to approach that of the unrestricted control marsh. As stated above, vegetation of the tide-restored marsh after 2 and 5 years of tide restoration remained significantly different from the unrestricted control marsh (Table 3); nevertheless, it is possible to demonstrate the current trajectory of vegetation change. To do this, a measure of dissimilarity based on the Euclidean distance metric (D_{max}) was calculated (Table 5). As D_{max} diminishes toward zero the unrestricted and tide-restored marshes are becoming more similar. In 1997 with no tide restoration and in 2000 with just limited tide restoration, dissimilarity between the marshes remained high as reflected in D_{max}. However, after 5 years of tide restoration D_{max} diminished toward zero, confirming that the vegetation is changing in an expected direction (i.e., reduction in brackish and woody species).

Table 5. D_{max}, an overall measure of dissimilarity between the unrestricted and tide-restricted or tide-restored marsh, demonstrates that the tide-restored marsh is becoming more similar to the unrestricted control marsh. As D_{max} diminishes toward zero the marshes are becoming more similar.

Comparisons	**Dissimilarity (D_{max})**
Unrestricted 97 vs. Tide-restricted 97	32
Unrestricted 00 vs. Tide-restored 00	35
Unrestricted 02 vs. Tide-restored 02	16

At this early stage of tidal restoration it is impossible to predict if the tide-restored marsh vegetation will ever be statistically similar to the unrestricted control marsh. Because the tide-restored marsh is located upstream of the unrestricted control marsh, similarity may not be achieved; but it is expected that the vegetation of the restored marsh will change in a trajectory toward the control marsh. Therefore, the trajectory of change becomes an important indicator of restoration success.

In addition to ANOSIM, the Hatches Harbor marsh vegetation data were analyzed by the ordination technique, Detrended Correspondence Analysis (DCA). Kent and Coker (1992) and Roman et al. (2001) provide detailed descriptions of ordination techniques including DCA. Ordination represents an objective means of ordering or arranging a multidimensional vegetation data set into fewer dimensions (*e.g.*, a 2-axis plot) such that any pattern that the data possess becomes more apparent. Figure 7 is a DCA ordination plot, with each point on the plot representing an individual 1-m^2 quadrat collected from the 1997 tide-restricted, 2000 tide-restored, or 2002 tide-restored marsh. Each quadrat has a different species composition and abundance resulting in a very complex data set, but as noted from the DCA diagram this complexity is arranged along a gradient. Quadrats with similar vegetation (in terms of species composition and abundance) are plotted close together, while dissimilar quadrats are far apart. On the far right side of the ordination diagram are quadrats composed of low marsh cover types (*e.g.*, *S. alterniflora*, open water). On the very left side there is a large grouping of quadrats primarily composed of woody and brackish wetland species. The quadrats at opposing ends of the plot are very different, with a gradient of communities in between, including *Phragmites* marsh and high marsh (e.g., *S. patens*).

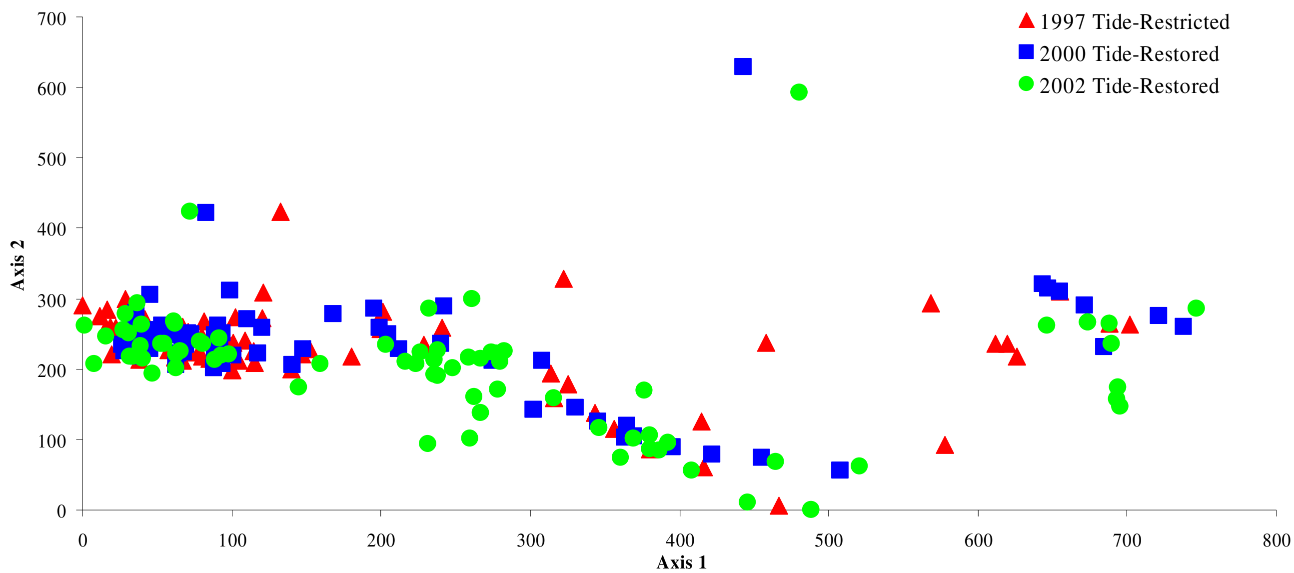


Figure 7. Ordination diagram by Detrended Correspondence Analysis of vegetation quadrats from Hatches Harbor tide-restricted/tide restored marsh. Individual 1-m^2 plots are shown. The plots fall along a gradient from woody and brackish vegetation to the distinctly different Low Marsh characterized by *S. alterniflora* and open water cover types. Also shown for some individual plots are time-trajectory lines (arrows) indicating change in the vegetation of the identical plot from 1997 to 2000 to 2002. Three plots are tracked over time (solid arrow, plot 4B160; coarse dash, plot 1B160; light dash, plot 1A40).

In terms of documenting temporal trends or trajectories in vegetation, ordination techniques, like DCA, are especially valuable tools (Austin 1977). Time trajectories (1997 pre-restoration to 2000 post-tidal restoration to 2002 post-tidal restoration) for several individual vegetation quadrats are plotted and it is clearly demonstrated that the vegetation is changing as expected. Plot 4B160, located 160m from the tidal creek of the tide-restored marsh, was characterized by *Rubus* and *Juncus effusus* in 1997, changed little after 2 years of tide restoration in 2000, but after 5 years of tide restoration (2002) the plot was dominated by *S. patens*, clearly demonstrating a shift in vegetation as expected. Plot 1A40, just 40 m from the creek and exposed to greater tidal influences, changed from *Juncus gerardii/S. patens* in 1997 to *S. alterniflora*-dominated in 2002. It is also noted that some plots, like 1B160 changed less dramatically, but this *Phragmites*-dominated plot in 1997 was characterized by dead *Phragmites* in 2000 and 2002. Application of multivariate vegetation analysis techniques, like ordination, will be instrumental in documenting these changes and better understanding the processes associated with vegetation recovery.

TRENDS IN *PHRAGMITES AUSTRALIS* POPULATIONS

Methods

Percent area-cover by plant species was determined using the point-intercept method (Roman et al. 2001) in July-August of 1998, 2000, and 2002. In October 1998, the stems of all *Phragmites australis* plants in transect 2 plots were counted and their heights measured to the nearest cm. The same data were collected in October 2002, but in all plots (i.e., transects 1-9). Live (growing season) *P. australis* biomass was then estimated from these parameters based on regressions developed by Thursby et al. (2002).

Cover

From 1998 to 2002, % cover of *P. australis* decreased in 32 of 103 plots in the tidally restricted portion of the marsh (i.e., transects 1-6). However, an increase in cover was observed in 27 plots. Moreover, the mean values of both positive and negative change were similar, suggesting zero net change in the cover of *P. australis* from a whole marsh perspective.

Trend	No. plots	Mean value of change (% cover)
no change	0	
negative	32	-32
positive	27	+27

Further analysis, however, revealed a strong spatial component underlying cover changes along individual transects. Cover tended to decline in plots closer to the main tidal creek, but increased or changed very little in plots distant from the creek (Figure 8). In general, zones of high *P. australis* cover shifted up-gradient, away from the influence of tide water. The most notable exceptions to this pattern occurred at 0 m (creek edge), where the positive value is due solely to an increase in *P. australis* cover at plot 6A/B-0, which is the furthest plot from the culverts in that distance category and receives much lower inputs of seawater.

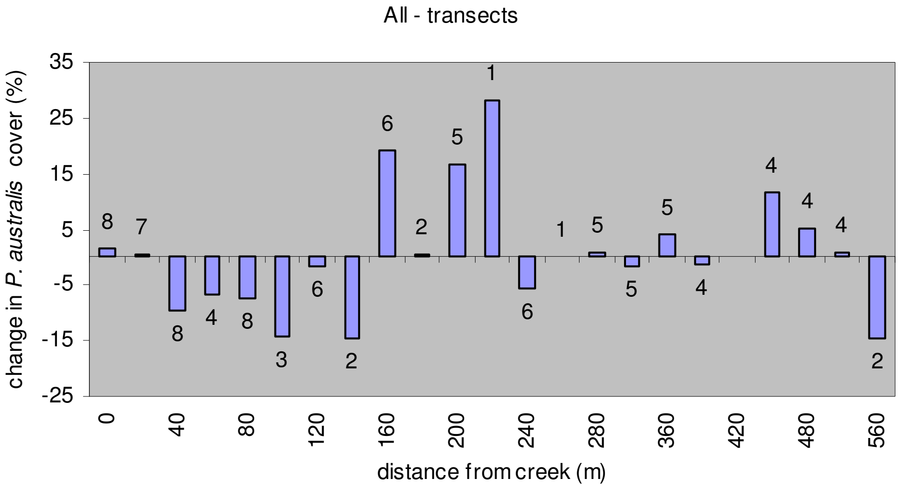


Figure 8. Average percent change from 1998 to 2002 in *P. australis* cover with distance away from the tidal creek (numbers above histograms represent no. of plots in each distance category).

Biomass

The mean estimated biomass of *P. australis* along transect 2 (all plots pooled) was significantly lower in 2002 than in 1998 ($F= 73.8$; $p<0.001$; T-test) (Figure 9). When all data were plotted against distance from the creek, no clear relationship emerged. When analyzed as separate transects, however, it became evident that distance was a significant correlate. For example, plots ranging between 0 and 100 m from the creek frequently exhibited large reductions in biomass while plots further away (>100 m) showed relatively small increases, decreases, or remained unchanged (Figure 10). Two-way ANOVA of stem height data along transect 2 (for which there were multiple values for each plot) similarly demonstrated that both time ($F=140.7$, $p<0.001$) and distance ($F=25.5$, $p<0.001$) exerted significant independent and interactive effects on *P. australis* biomass. In this regard, major reductions in stem height occurred in plots between 0 and 100 m while more distant plots showed comparatively minor variation or static responses.

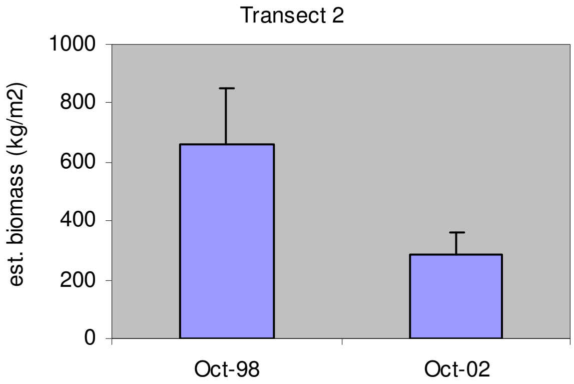


Figure 9. Mean biomass of *P. australis* in 1998 and 2002 (all plots, all transects)

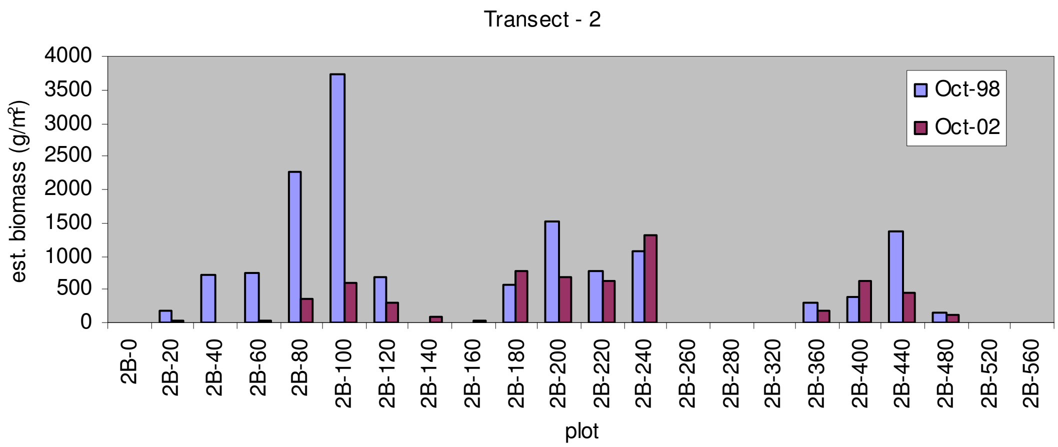
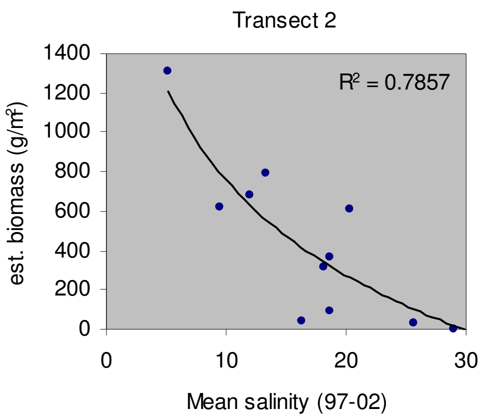


Figure 10. Biomass of *P. australis* in 1998 and 2002 along transect 2.

Influence of physico-chemical factors

Porewater samples collected from a subset of plots along transect 2 were analyzed for salinity, nutrients, pH, conductivity, iron, and sulfides. Plot elevations were determined in a 1998 survey and depth to water during low tide over the spring neap period was recorded in 2002. Multiple regression of biomass against these parameters revealed that the average 4-year salinity value (1998-2002), average depth to water during spring neap tide, and elevation comprised the best subset of predictor variables. In this relationship, both mean salinity and elevation were negatively correlated with biomass while depth to water was positively correlated (BIO = 4436 - 69*[mean salinity] + 344*[depth to water] -2328*[elevation]; r^2=0.77). However, a curvilinear regression of biomass vs. mean salinity produced a higher r^2 value (0.78), suggesting that long-term salinity conditions can be used alone as a reliable indicator of *P. australis* vigor in this system (Figure 11a). Mean stem height, the heights of the tallest 5 stems, and percentage of stems bearing an inflorescence also showed good inverse relationships with salinity (Figure 11b,c)



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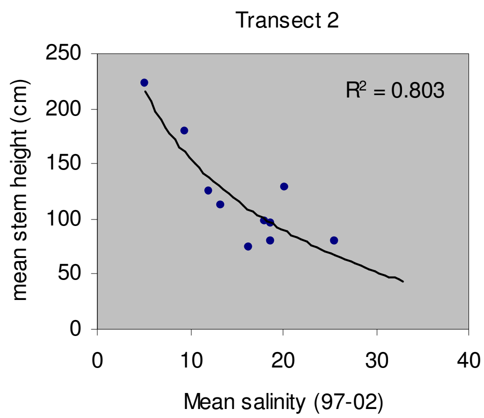

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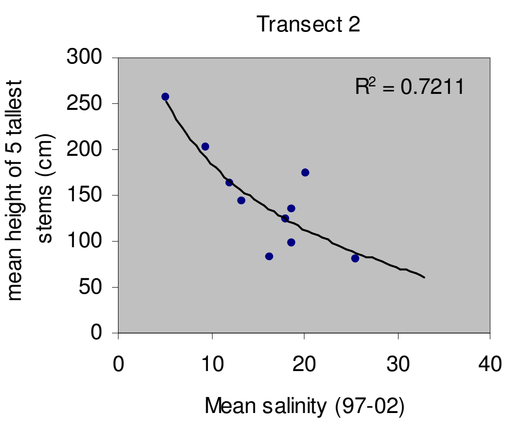

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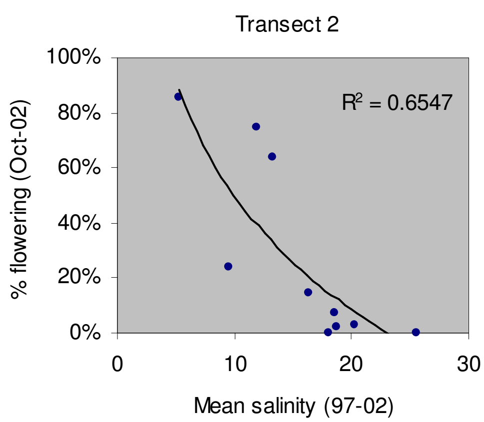


D.

Figure 11. Regressions of a) biomass, b) mean stem height, c) mean height of 5 tallest stems, and d) % flowering stems of *P. australis* vs. mean salinity (98-02).

Discussion

An important finding of this work is that the overall vigor (as indicated by estimates of standing live biomass) of *P. australis* is not well represented by point-intercept cover or stem densities. Cover and stem densities also were poorly correlated with the salinity gradient along transect 2, in contrast to stem height and biomass (Figure 11). While measures of abundance in the horizontal plane provide very useful information on community composition and changes therein, they do not characterize the vertical structure of vegetation - a critical aspect of *P. australis* stands given their ability to attain heights exceeding 3 m under optimal growing conditions. Accordingly, measurements of stem height should be incorporated into all future monitoring of this species.

Determining percentages of stems that produce an inflorescence may also prove helpful in gauging the physiological state of *P. australis* (Figure 11d).

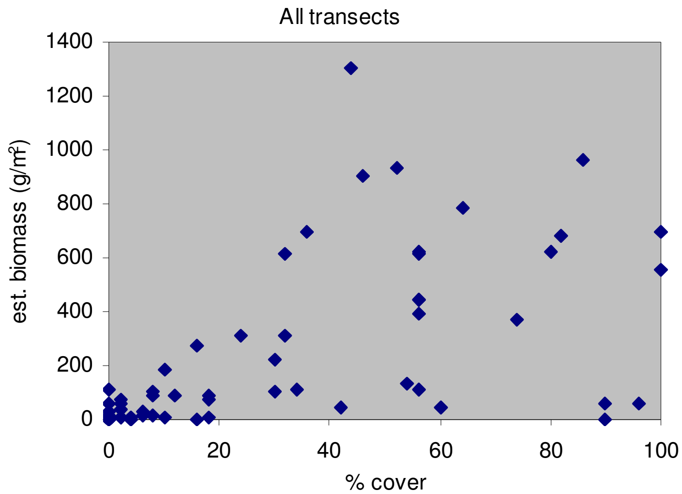


Figure 12. *P. australis* biomass (from Oct-02) plotted against % cover (Jul/Aug-02).

During the time that Hatches Harbor has undergone tidal restoration, prolonged periods of elevated salinities (Fig. 5) appear to have resulted in extreme osmotic stress in the roots of *P. australis*. This stress is manifested as significant reductions in stand height and biomass, particularly in areas adjacent to the main tidal creek and near the culverts. In certain places that are distant from both the main tidal creek and the culverts, *P. australis* still survives in dense stands. Presumably this is due to low levels of tidal influence (and therefore low salinities) as a function of distances from the source of full-strength seawater. Although salinities may be substantially higher compared to pre-restoration levels, they are not yet high enough (for long enough) to adversely affect *P. australis*. In addition, *P. austalis* may be benefiting from reduced interspecific competition as many co-existing salt-intolerant species have since disappeared.

P. australis reportedly compensates for high salinities by enhancing proline synthesis in the roots (Hartzendorf & Rolletschek 2001). Given the low levels of sulfide measured in porewater samples (Fig. 6), it is unlikely that *P. australis* is suffering from sulfide toxicity and associated problems with nitrogen assimilation. In high salinity areas, the demand for osmolyte production and increased respiration may explain diminishing aboveground biomass. Photosynthate and reserve carbohydrates (starch) are likely being depleted by root metabolism at the expense of vertical shoot growth and sexual reproduction. Leaf and rhizome tissue samples are currently being analyzed in the North Atlantic Coastal Laboratory in an effort to evaluate these hypotheses.

MOSQUITOES

The species composition and abundance of floodwater-breeding in coastal marshes is affected by salinity, flooding duration and areal extent, and access to breeding sites by (primarily fish) predators. The objective of mosquito monitoring at Hatches Harbor is to assess whether changes in hydrography caused by the tidal restoration affect mosquito species composition and abundance.

Adult mosquitoes have been monitored throughout July and August from 1997 through 2002 using light/CO_2 traps set at least biweekly in three general habitats within the Hatches Harbor flood plain (Fig. 1): at the dike, at the end of the taxiway, and at the airport terminal. The dike station is surrounded by saline breeding habitat. The taxiway was until 1999 surrounded by seasonally flooded freshwater wetlands; where since April 2001 salinity and flooding frequency have increased sharply. Trapping at the airport terminal represents the upstream and freshest end of the estuarine gradient; it is the location where most people and mosquitoes meet, i.e. the most likely location within the system of a mosquito nuisance. Two traps are set in each location in the late afternoon and retrieved in early morning. Trapped adult mosquitoes are identified and counted by species.

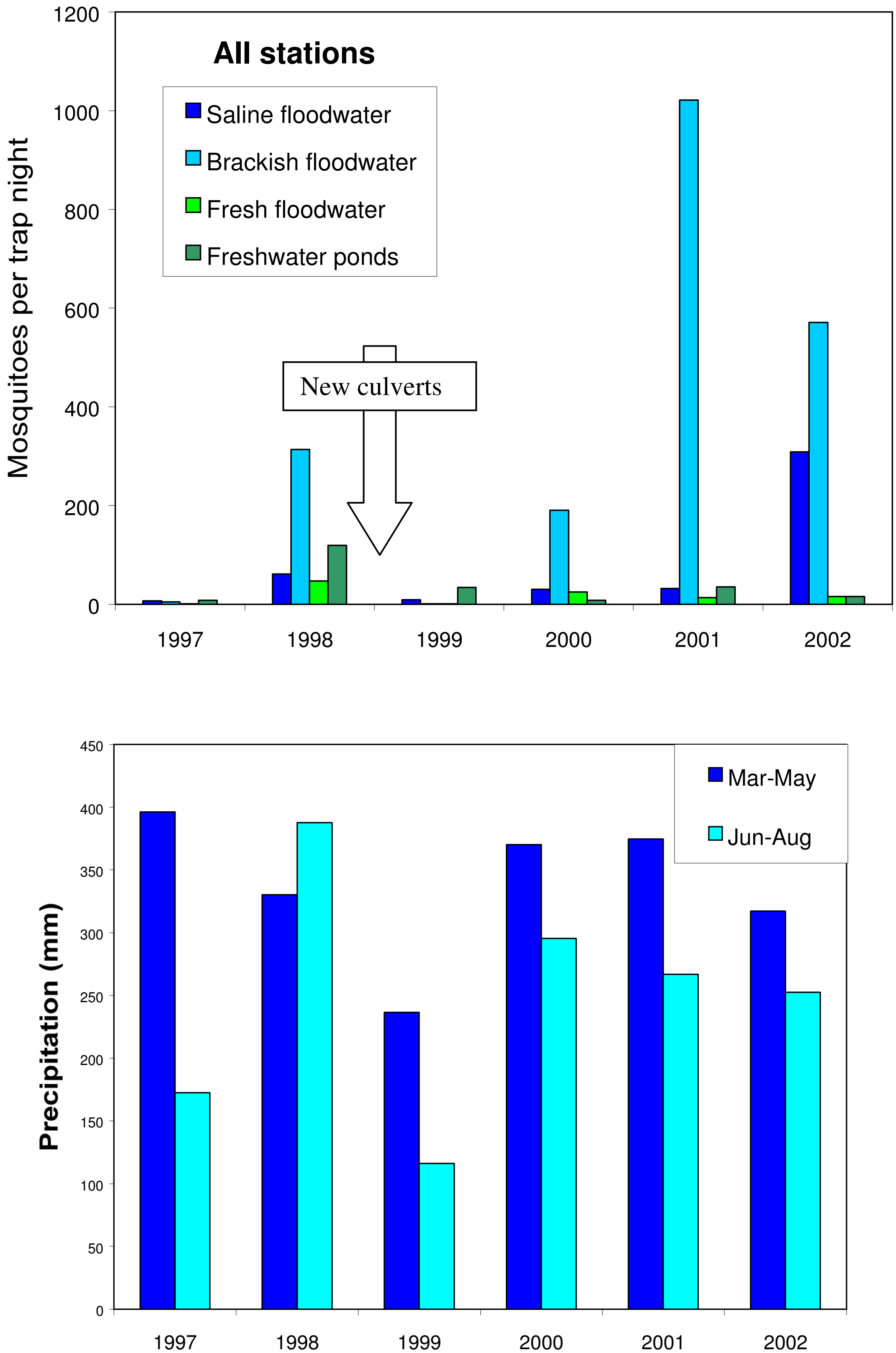
Prior to 2001, adult mosquitoes were predominantly brackish- and freshwater-breeding species whose annual abundance appeared to correlate with precipitation, especially Jun-Aug rainfall (Fig. 13). This is consistent with tide height observations (Fig. 3) that indicate little tidal flooding of the wetland surface; i.e. floodwater breeding before 2001 was dependent on precipitation, not tidal flooding. This mode of flooding would result in low surface-water salinity, thus, the preponderance of brackish and freshwater, rather than salt marsh, mosquitoes (*Ochlerotatus sollicitans*).

In April 2001 effective culvert opening was increased four fold causing increased tide heights (Fig. 3); consequently, much more water of much higher salinity reached and, in places, remained for weeks on the wetland surface. This appears to have caused a qualitative change in mosquito breeding ecology: in that summer , very high numbers of brackish-water breeders (*Ochlerotatus cantator*) were captured. In 2002, high captures of both brackish and saltwater-breeding species (*O. sollicitans*) followed still more extensive and higher-salinity flooding (Fig. 13).

Statistical comparisons (Analysis of Variance, P=0.05) of inter-annual trap collections by habitat and species show significant increases in brackish-breeding species (*O. cantator*) at the airport taxiway and terminal stations in 2001, and in saltwater-breeding species (*O. sollicitans)* at the taxiway and dike stations in 2002. This sequence of brackish- and then saltwater-breeding mosquitoes corresponded with the step-wise increase in the extent of seawater flooding of the wetland surface. Species that breed in freshwater habitats (*Coquilletidia perturbans* and *O. canadensis*) have significantly declined since 1998.

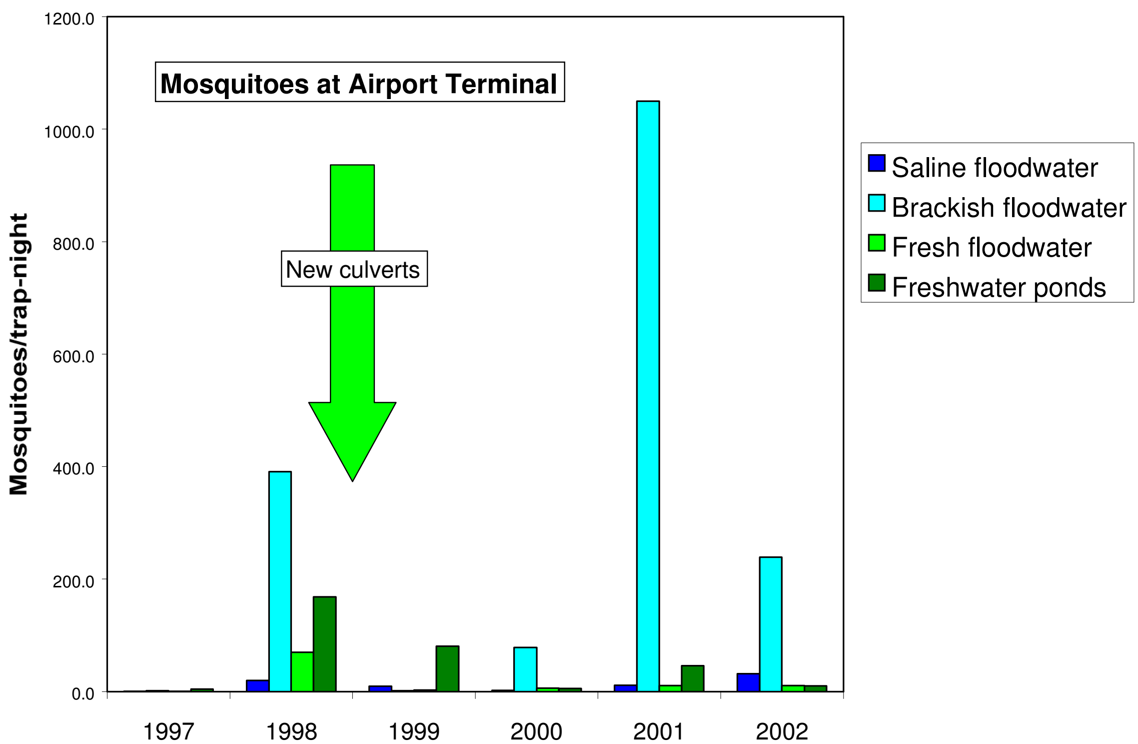
Specific breeding sites within the Hatches Harbor flood plain are being studied by the Cape Cod Mosquito Control Project (CCMCP), who will report on findings at the December 2002 Technical Review Committee meeting; however, adult trapping results and casual observations of the occurrence of chronic standing water suggest that most intensive breeding is in low areas just seaward (southwest) of the Airport taxiway. Trap

Figure 13. Hatches Harbor mosquito collections by breeding habitat (top panel) and outer Cape Cod spring-summer precipitation, 1997-2002.



counts are nearly always highest at the end of the taxiway. The wetland in this area is typically flooded for long periods between spring tides. Floodwater is often topographically isolated ("land-locked") hindering predatory fish access via tidal creeks. Importantly, the topography of this area has very likely been artificially altered by subsidence and dune migration during the nearly 70 years of blocked tidal flow (1930 to 1999); therefore, high mosquito production may be an artifact of human disturbance to wetland hydrography and sedimentation. Historical aerial photographs (1938, 1947) seem to show a shallow creek system connecting this area with the main tidal creek; with the dike's elimination of surface tidal flow, it appears that the old tributary creek has since filled with wind-blown sand. With this perspective, it may be consistent with restoration project goals to re-establish creeks ditch through these low swales to improve, and probably restore, tidal exchange and fish access (Tyler & Zieman 1999, Williams et al. 2002). The remoteness of this area from human habitation argues against creek construction strictly for the purpose of nuisance mosquito control. Moreover, note that brackish species were abundant at the Airport terminal in 2001, but less so in 2002; salt marsh mosquitoes were scarce here in both years (Fig. 14).

Figure 14. Mosquito collections at the Provincetown Airport Terminal area, 1997-2002.



CONCLUSIONS

Tide heights

- Tide heights and tidal range have increased substantially in the restricted marsh without exceeding the critical flooding threshold at the Airport.
- Tide heights in the restoring marsh are now about 60% those of unrestricted Hatches Harbor.
- Low tide elevations remain low, facilitating peat drainage at low tide.

Sediment-water quality

- Nearly full-strength seawater (30 ppt) is penetrating typically 200 m into previously *Phragmites*-dominated plant communities.
- Porewater sulfides remain extremely low, are not likely to affect plant growth, nor to increase given continued efficient drainage, and can therefore probably be discontinued from regular monitoring.

Vegetation

Based on an analysis of over 100 permanent 1-m^2 vegetation plots, sampled before tidal restoration in 1997, and then 2 and 5 growing seasons after tidal restoration (2000 and 2002, respectively), the Hatches Harbor marsh is responding to increased tidal flow as expected.

- Woody (e.g., *Myrica*), thicket (e.g., *Rubus*) and brackish-water species (e.g., *Lythrum*, *Juncus effuses*) are decreasing in relative abundance throughout the tide-restoring marsh, resulting in a statistically different vegetation community from 1997 tide-restricted to 2002 tide-restored.
- Vegetation of the tide-restored marsh remains quite different from the unrestricted control marsh; however, it is demonstrated that the tide-restored vegetation is changing in a direction or trajectory that is similar to the *Spartina*-dominated control marsh.
- Salinity stress to *Phragmites* is manifested as significant reductions in stand height and biomass, particularly in areas adjacent to the main tidal creek and near the culverts.
- In certain places that are distant from both the main tidal creek and the culverts, *P. australis* still survives in dense stands.

The restoration process at Hatches Harbor is in the initial stages and it may take decades for the complete benefits of tidal restoration to be realized, but it is encouraging to note that vegetation is changing as expected. Monitoring vegetation by the permanent plot method is tedious, but the method enables investigators to track very subtle changes in vegetation and to quantify those changes with statistical certainty. Permanent plot vegetation monitoring should continue at 5-yr intervals, thereby producing a data set that will enhance our understanding of the marsh restoration process, increase our ability to predict vegetation responses to tidal restoration, and provide useful information for coastal managers engaged in marsh restoration at other sites.

Mosquitoes

- Brackish and salt marsh mosquito production has increased as higher high tides have increased flooding extent and salinity on the wetland surface.
- Increased breeding appears to be focused on swales that flood with tidal water just southwest of the Airport taxiway. Because these swales may be an artifact of diking, creek construction to improve tidal exchange is an adaptive management option that may improve tidal exchange, expedite habitat restoration, ease fish access and reduce floodwater mosquito production.

PLANS FOR 2003

- Continue tide height, sediment-water salinity and mosquito monitoring.
- Repeat nekton monitoring conducted previously in 1997-8, before restoration, and in 1999. We anticipate a major change in nekton use of the marsh since tide heights and the extent of wetland flooding were increased dramatically in 2001.
- Collect sediment cores and quantitatively describe physical and chemical peat characteristics at vegetation plots.
- Continue to monitor *Phragmites* survival and vigor in the restricted, now restoring, marsh.

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